CHAPTER 10

RADIATION RISK ASSESSMENT GUIDANCE

There are many sites contaminated with radioactive substances that are included on the National Priorities List (NPL), and additional sites are expected in future NPL updates. This chapter provides supplemental baseline risk assessment guidance for use at these sites. This guidance is intended as an overview of key differences in chemical and radionuclide assessments, and not as a comprehensive, stand-alone approach for assessing the risks posed by radiation.

The reader should be familiar with the guidance provided in Chapters 2 through 9 before proceeding further in Chapter 10. Although the discussions in the previous chapters focus primarily on chemically contaminated sites, much of the information presented is also applicable to the evaluation of radioactively contaminated Superfund sites. For consistency and completeness, the topics discussed in each section of this chapter parallel the topics covered in each of the previous chapters.

After a brief introduction to some of the basic principles and concepts of radiation protection (Section 10.1), seven additional areas are addressed:

(1) Regulation of Radioactively Contaminated Sites (Section 10.2);

(2) Data Collection (Section 10.3);

(3) Data Evaluation (Section 10.4);

(4) Exposure and Dose Assessment (Section 10.5);

ACRONYMS, SYMBOLS, AND UNITS FOR CHAPTER 10

A(t) = Activity at Time t
Bq = Becquerel
Ci = Curie
CLP = Contract Laboratory Program
D = Absorbed Dose
DCF = Dose Conversion Factor Per Unit Intake
H_E = Effective Dose Equivalent
H_T = Dose Equivalent Averaged Over Tissue or Organ T
$H_{E,50}$ = Committed Effective Dose Equivalent Per Unit Intake
$H_{T,50}$ = Committed Dose Equivalent Averaged Over Tissue T
LET = Linear Energy Transfer
LLD = Lower Limit of Detection
MeV = Million Electron Volts
N = Modifying Factor in the Definition of Dose Equivalent
pCi = PicoCurie (10^{-12} Ci)
Q = Quality Factor in Definition of Dose Equivalent
RBE = Relative Biological Effectiveness
SI = International System of Units
Sv = Sievert
T = Tissue or Target Organs
w_T = Weighting Factor in the Definition of Effective Dose Equivalent and Committed Effective Dose Equivalent

(5) Toxicity Assessment (Section 10.6);

(6) Risk Characterization (Section 10.7); and

(7) Documentation, Review, and Management Tools for the Risk Assessor, Reviewer, and Manager (Section 10.8).

DEFINITIONS FOR CHAPTER 10

Absorbed Dose (D). The mean energy imparted by ionizing radiation to matter per unit mass. The special SI unit of absorbed dose is the gray (Gy); the conventional unit is the rad (1 rad = 0.01 Gy).

Becquerel (Bq). One nuclear disintegration per second; the name for the SI unit of activity. 1 Bq = 2.7 x 10^{-11} Ci.

Committed Dose Equivalent ($H_{T,50}$). The total dose equivalent (averaged over tissue T) deposited over the 50-year period following the intake of a radionuclide.

Committed Effective Dose Equivalent ($H_{E,50}$). The weighted sum of committed dose equivalents to specified organs and tissues, in analogy to the effective dose equivalent.

Curie (Ci). 3.7 x 10^{10} nuclear disintegrations per second, the name for the conventional unit of activity. 1 Ci = 3.7 x 10^{10}Bq.

Decay Product(s). A radionuclide or a series of radionuclides formed by the nuclear transformation of another radionuclide which, in this context, is referred to as the parent.

Dose Conversion Factor (DCF). The dose equivalent per unit intake of radionuclide.

Dose Equivalent (H). The product of the absorbed dose (D), the quality factor (Q), and any other modifying factors (N). The SI unit of dose equivalent is the sievert (Sv); the conventional unit is the rem (1 rem = 0.01 Sv).

Effective Dose Equivalent (H_E). The sum over specified tissues of the products of the dose equivalent in a tissue or organ (T) and the weighting factor for that tissue.

External Radiation. Radiations incident upon the body from an external source.

Gray (Gy). The SI unit of absorbed dose. 1Gy = 1 Joule kg^{-1} = 100 rad.

Half-Life (physical, biological, or effective). The time for a quantity of radionuclide, i.e., its activity, to diminish by a factor of a half (because of nuclear decay events, biological elimination of the material, or both.).

Internal Radiation. Radiation emitted from radionuclides distributed within the body.

Ionizing Radiation. Any radiation capable of displacing electrons from atoms or molecules, thereby producing ions.

Linear Energy Transfer (LET). A measure of the rate of energy absorption, defined as the average energy imparted to the absorbing medium by a charged particle per unit distance (KeV per um).

Nuclear Transformation. The spontaneous transformation of one radionuclide into a different nuclide or into a different energy state of the same nuclide.

Quality Factor (Q). The principal modifying factor that is employed in deriving dose equivalent, H, from absorbed dose, D; chosen to account for the relative biological effectiveness (RBE) of the radiation in question, but to be independent of the tissue or organ under consideration, and of the biological endpoint. For radiation protection purposes, the quality factor is determined by the linear energy transfer (LET) of the radiation.

Rad. The conventional unit for absorbed dose of ionizing radiation; the corresponding SI unit is the gray (Gy); 1 rad = 0.01 Gy = 0.01 Joule/kg.

Rem. An acronym of radiation equivalent man, the conventional unit of dose equivalent; the corresponding SI unit is the Sievert; 1 Sv = 100 rem.

Sievert (Sv). The special name for the SI unit of dose equivalent. 1 Sv = 100 rem.

Slope Factor. The age-averaged lifetime excess cancer incidence rate per unit intake (or unit exposure for external exposure pathways) of a radionuclide.

Weighting Factor (w_T). Factor indicating the relative risk of cancer induction or hereditary defects from irradiation of a given tissue or organ; used in calculation of effective dose equivalent and committed effective dose equivalent.

There are special hazards associated with handling radioactive waste and EPA strongly recommends that a health physicist experienced in radiation measurement and protection be consulted prior to initiating any activities at a site suspected of being contaminated with radioactive substances. EPA also recommends that the remedial project manager (RPM) or on-scene coordinator (OSC) should designate both a chemical risk assessor and a radiation risk assessor. These individuals should work closely with each other and the RPM to coordinate remedial activities (e.g., site scoping, health and safety planning, sampling and analysis) and exchange information common to both chemical and radionuclide assessments, including data on the physical characteristics of the site, potentially impacted populations, pathways of concern, and fate and transport models used. At the conclusion of the remedial investigation/feasibility study (RI/FS) process, the RPM should issue a single report that summarizes and integrates the results from both the chemical and the radiation risk assessments.

A two-phase evaluation is described for the radiation risk assessment. As discussed in Section 10.5, procedures established by the International Commission on Radiological Protection (ICRP 1979) and adopted by EPA in *Federal Guidance Report No. 11* (EPA 1988) are used to estimate the radiation dose equivalent to humans from potential exposures to radionuclides through all pertinent exposure pathways at a site. Those estimates of dose equivalent may be used for comparison with established radiation protection standards and criteria. However, this methodology was developed for regulation of occupational radiation exposures for adults and is not completely applicable for estimating health risk to the general population at a Superfund site. Therefore, a separate methodology is presented in Section 10.7.2 for estimating health risk, based on the age-averaged lifetime excess cancer incidence per unit intake (and per unit external exposure) for radionuclides of concern. Radiation risk assessments for Superfund sites should include estimates of both the dose equivalent computed as described in Section 10.5, and the health risk attributable to radionuclide exposures computed using the approach described in Section 10.7.

Only summary-level information is presented in this chapter, and references are provided to a number of supporting technical documents for further information. In particular, the reader is encouraged to consult Volume 1 of the *Background Information Document for the Draft Environmental Impact Statement for Proposed NESHAPS for Radionuclides* (EPA 1989a) for a more comprehensive discussion of EPA's current risk assessment methodology for radionuclides.

For additional radiation risk assessment information and guidance, RPMs and other interested individuals can contact the Office of Radiation Programs (ORP) within EPA headquarters at 202-475-9630 (FTS 475-9630). Interested individuals also can contact the Regional Radiation Program Managers within each of the EPA regional offices for guidance and health physics support.

10.1 RADIATION PROTECTION PRINCIPLES AND CONCEPTS

Radioactive atoms undergo spontaneous nuclear transformations and release excess energy in the form of ionizing radiation. Such transformations are referred to as radioactive decay. As a result of the radioactive decay process, one element is transformed into another; the newly formed element, called a decay product, will possess physical and chemical properties different from those of its parent, and may also be radioactive. A radioactive species of a particular element is referred to as a radionuclide or radioisotope. The exact mode of radioactive transformation for a particular radionuclide depends solely upon its nuclear characteristics, and is independent of the nuclide's chemical characteristics or physical state. A fundamental and unique characteristic of each radionuclide is its radioactive half-life, defined as the time required for one half of the atoms in a given quantity of the radionuclide to decay. Over 1,600 different radionuclides have been identified to date, with half-lives ranging from fractions of a second to millions of years. Selected radionuclides of potential importance at Superfund sites are listed in Exhibit 10-1.

Radiation emitted by radioactive substances can transfer sufficient localized energy to atoms to remove electrons from the electric field of their nucleus (ionization). In living tissue this energy transfer can destroy cellular constituents and produce electrically charged molecules (i.e., free radicals). Extensive biological damage can lead to adverse health effects. The type of ionizing radiation emitted by a particular radionuclide depends upon the exact nature of the nuclear transformation, and may include emission of alpha particles, electrons (beta particles or positrons), and neutrons; each of these transformations may be accompanied by emission of photons (gamma radiation or x-rays). Each type of radiation differs in its physical characteristics and in its ability to inflict damage to biological tissue. These characteristics and effects are summarized in the box on this page.

Quantities of radionuclides are typically expressed in terms of activity at a given time t (A(t)). The SI unit of activity is the becquerel (Bq), which is defined as the quantity of a given radionuclide in which one atom is transformed per second (i.e., one decay per second). The conventional unit of activity is the curie (Ci), which is defined as the quantity of a given radionuclide in which $3.7x10^{10}$ atoms undergo nuclear transformation each second; one curie is approximately equivalent to the decay rate of one gram of Ra-226. A more convenient unit of activity for expressing environmental concentrations of radionuclides is the picoCurie (pCi), which is equal to 10^{-12} Ci. Occasionally, activity is expressed incorrectly in terms of counts per second (cps) or counts per minute (cpm): these refer to the number of transformations per unit time measured by a particular radiation detector and do not represent the true decay rate of the radionuclide. To derive activity values, count rate measurements are multiplied by radioisotope-specific detector calibration factors.

PRINCIPAL TYPES OF IONIZING RADIATION

Alpha particles are doubly charged cations, composed of two protons and two neutrons, which are ejected monoenergetically from the nucleus of an atom when the neutron to proton ratio is too low. Because of their relatively large mass and charge, alpha particles tend to ionize nearby atoms quite readily, expending their energy in short distances. Alpha particles will usually not penetrate an ordinary sheet of paper or the outer layer of skin. Consequently, alpha particles represent a significant hazard only when taken into the body, where their energy is completely absorbed by small volumes of tissues.

Beta particles are electrons ejected at high speeds from the nucleus of an unstable atom when a neutron spontaneously converts to a proton and an electron. Unlike alpha particles, beta particles are not emitted with discrete energies but are ejected from the nucleus over a continuous energy spectrum. Beta particles are smaller than alpha particles, carry a single negative charge, and possess a lower specific ionization potential. Unshielded beta sources can constitute external hazards if the beta radiation is within a few centimeters of exposed skin surfaces and if the beta energy is greater than 70 keV. Beta sources shielded with certain metallic materials may produce bremsstrahlung (low energy x-ray) radiation which may also contribute to the external radiation exposure. Internally, beta particles have a much greater range than alpha particles in tissue. However, because they cause fewer ionizations per unit path length, beta particles deposit much less energy to small volumes of tissue and, consequently, inflict must less damage than alpha particles.

Positrons are identical to beta particles except that they have a positive charge. A positron is emitted from the nucleus of a neutron-deficient atom when a proton spontaneously transforms into a neutron. Alternatively, in cases where positron emission is not energetically possible, the neutron deficiency may be overcome by electron capture, whereby one of the orbital electrons is captured by the nucleus and united with a proton to form a neutron, or by annihilation radiation, whereby the combined mass of a positron and electron is converted into photon energy. The damage inflicted by positrons to small volumes of tissue is similar to that of beta particles.

Gamma radiations are photons emitted from the nucleus of a radioactive atom. X-rays, which are extra-nuclear in origin, are identical in form to gamma rays, but have slightly lower energy ranges. There are three main ways in which x- and gamma rays interact with matter: the photoelectric effect, the Compton effect, and pair production. All three processes yield electrons which then ionize or excite other atoms of the substance. Because of their high penetration ability, x- and gamma radiations are of most concern as external hazards.

Neutrons are emitted during nuclear fission reactions, along with two smaller nuclei, called fission fragments, and beta and gamma radiation. For radionuclides likely to be encountered at Superfund sites, the rate of spontaneous fission is minute and no significant neutron radiation is expected.

EXHIBIT 10-1

RADIOLOGICAL CHARACTERISTICS OF SELECTED RADIONUCLIDES FOUND AT SUPERFUND SITES[a]

Nuclide	Half-life[c]	Average Radiation Energies (MeV/decay)[b]		
		Alpha	Beta, Electron	x, Gamma
Am-241	4.32×10^{2} y	5.57×10^{0}	5.21×10^{-2}	3.25×10^{-2}
Am-243	7.38×10^{3} y	5.36×10^{0}	2.17×10^{-2}	5.61×10^{-2}
Ba-137m	2.55×10^{0} h	--	6.37×10^{-2}	5.98×10^{-1}
C-14	5.73×10^{3} y	--	4.95×10^{-2}	--
Ce-144	2.84×10^{2} d	--	9.22×10^{-2}	2.07×10^{-2}
Cm-243	2.85×10^{1} y	5.89×10^{0}	1.38×10^{-1}	1.35×10^{-1}
Cm-244	1.81×10^{1} y	5.89×10^{0}	8.59×10^{-3}	1.70×10^{-3}
Co-60	5.27×10^{0} y	--	9.65×10^{-2}	2.50×10^{0}
Cr-51	2.77×10^{1} d	--	3.86×10^{-3}	3.26×10^{-2}
Cs-134	2.06×10^{0} y	--	1.64×10^{-1}	1.55×10^{0}
Cs-135	2.30×10^{6} y	--	6.73×10^{-2}	--
Cs-137	3.00×10^{1} y	--	1.87×10^{-1}	--
Fe-59	4.45×10^{1} d	--	1.17×10^{-1}	1.19×10^{0}
H-3	1.23×10^{1} y	--	5.68×10^{-3}	--
I-129	1.57×10^{7} y	--	6.38×10^{-2}	2.46×10^{-2}
I-131	8.04×10^{0} d	--	1.92×10^{-1}	3.81×10^{-1}
K-40	1.28×10^{9} y	--	5.23×10^{-1}	1.56×10^{-1}
Mn-54	3.13×10^{2} d	--	4.22×10^{-3}	8.36×10^{-1}
Mo-99	6.60×10^{1} h	--	3.93×10^{-1}	1.50×10^{-1}
Nb-94	2.03×10^{4} y	--	1.68×10^{-1}	1.57×10^{0}
Np-237	2.14×10^{6} y	4.85×10^{0}	7.01×10^{-2}	3.46×10^{-2}
P-32	1.43×10^{1} d	--	6.95×10^{-1}	--
Pb-210	2.23×10^{1} y	--	3.80×10^{-2}	4.81×10^{-3}
Po-210	1.38×10^{2} d	5.40×10^{0}	8.19×10^{-8}	8.51×10^{-6}
Pu-238	8.77×10^{1} y	5.59×10^{0}	1.06×10^{-2}	1.81×10^{-3}
Pu-239	2.41×10^{4} y	5.24×10^{0}	6.74×10^{-3}	8.07×10^{-4}
Pu-240	6.54×10^{3} y	5.24×10^{0}	1.06×10^{-2}	1.73×10^{-3}
Pu-241	1.44×10^{1} y	1.22×10^{-4}	5.25×10^{-3}	2.55×10^{-6}
Pu-242	3.76×10^{5} y	4.97×10^{0}	8.73×10^{-3}	1.44×10^{-3}
Ra-226	1.60×10^{3} y	4.86×10^{0}	3.59×10^{-3}	6.75×10^{-3}
Ra-228	5.75×10^{0} y	--	1.69×10^{-2}	4.14×10^{-9}
Ru-106	3.68×10^{2} d	--	1.00×10^{-2}	--
S-35	8.74×10^{1} d	--	4.88×10^{-2}	--
Sr-89	5.05×10^{1} d	--	5.83×10^{-1}	8.45×10^{-5}
Sr-90	2.91×10^{1} y	--	1.96×10^{-1}	--
Tc-99	2.13×10^{5} y	--	1.01×10^{-1}	--
Tc-99m	6.02×10^{0} h	--	1.62×10^{-2}	1.26×10^{-1}
Th-230	7.70×10^{4} y	4.75×10^{0}	1.42×10^{-2}	1.55×10^{-3}
Th-232	1.41×10^{10} y	4.07×10^{0}	1.25×10^{-2}	1.33×10^{-3}
U-234	2.44×10^{5} y	4.84×10^{0}	1.32×10^{-2}	1.73×10^{-3}
U-235	7.04×10^{8} y	4.47×10^{0}	4.92×10^{-2}	1.56×10^{-1}
U-238	4.47×10^{9} y	4.26×10^{0}	1.00×10^{-2}	1.36×10^{-3}

[a] Source: ICRP 1983 (except Ba-137m data from Kocher 1981).
[b] Computed as the sum of the products of the energies and yields of individual radiations.
[c] Half-life expressed in years (y), days (d), and hours (h).

The activity per unit mass of a given radionuclide is called the specific activity, and is usually expressed in units of becquerels per gram (Bq/g) or curies per gram (Ci/g). The shorter the half-life of the radionuclide, the greater is its specific activity. For example, Co-60 has a radioactive half-life of about 5 years and a specific activity of $4x10^{13}$ Bq/g, whereas Np-237 has a half-life of 2 million years and a specific activity of $3x10^{7}$ Bq/g.

Several terms are used by health physicists to describe the physical interactions of different types of radiations with biological tissue, and to define the effects of these interactions on human health. One of the first terms developed was radiation exposure, which refers to the transfer of energy from a radiation field of x- or gamma rays to a unit mass of air. The unit for this definition of exposure is the roentgen (R), expressed as coulombs of charge per kilogram of air (1 R = $2.58x10^{-4}$ C/kg).

The term exposure is also defined as the physical contact of the human body with radiation. Internal exposure refers to an exposure that occurs when human tissues are subjected to radiations from radionuclides that have entered the body via inhalation, ingestion, injection, or other routes. External exposure refers to the irradiation of human tissues by radiations emitted by radionuclides located outside the body either dispersed in the air or water, on skin surfaces, or deposited on ground surfaces. All types of radiation may contribute to internal exposure, whereas only photon, beta, and neutron radiations contribute significantly to external exposure.

Ionizing radiation can cause deleterious effects on biological tissues only when the energy released during radioactive decay is absorbed in tissue. The absorbed dose (D) is defined as the mean energy imparted by ionizing radiation per unit mass of tissue. The SI unit of absorbed dose is the joule per kilogram, also assigned the special name the gray (1 Gy = 1 joule/kg). The conventional unit of absorbed dose is the rad (1 rad = 100 ergs per gram = 0.01 Gy).

For radiation protection purposes, it is desirable to compare doses of different types of radiation. The absorbed dose of any radiation divided by the absorbed dose of a reference radiation (traditionally 250 kVp x-rays) that produces the same biological endpoint is called the Relative Biological Effectiveness or RBE. For regulatory purposes, an arbitrary consensus RBE estimate called the Quality Factor or Q is often used. The dose equivalent (H) was developed to normalize the unequal biological effects produced from equal absorbed doses of different types of radiation. The dose equivalent is defined as:

$$H = DQN$$

where D is the absorbed dose, Q is a quality factor that accounts for the RBE of the type of radiation emitted, and N is the product of any additional modifying factors. Quality factors currently assigned by the International Commission on Radiological Protection (ICRP) include values of Q=20 for alpha particles, Q=10 for neutrons and protons, and Q=1 for beta particles, positrons, x-rays, and gamma rays (ICRP 1984). These factors may be interpreted as follows: on average, if an equal amount of energy is absorbed, an alpha particle will inflict approximately 20 times more damage to biological tissue than a beta particle or gamma ray, and twice as much damage as a neutron. The modifying factor is currently assigned a value of unity (N=1) for all radiations. The SI unit of dose equivalent is the sievert (Sv), and the conventional unit is the rem (1 rem = 0.01 Sv).

GENERAL HEALTH PHYSICS REFERENCES

Introduction to Health Physics (Cember 1983)

Atoms, Radiation, and Radiation Protection (Turner 1986)

Environmental Radioactivity (Eisenbud 1987)

The Health Physics and Radiological Health Handbook (Shleien and Terpilak 1984)

EFFECTIVE DOSE EQUIVALENT

The effective dose equivalent, H_E , is a weighted sum of dose equivalents to all organs and tissues (ICRP 1977, ICRP 1979), defined as:

$$H_E = \sum_T w_T H_T$$

where w_T is the weighting factor for organ or tissue T and H_T is the mean dose equivalent to organ or tissue T. The factor w_T, which is normalized so that the summation of all the organ weighting factors is equal to one, corresponds to the fractional contribution of organ or tissue T to the total risk of stochastic health effects when the body is uniformly irradiated. Similarly, the committed effective dose equivalent, $H_{E,50}$, is defined as the weighted sum of committed dose equivalents to all irradiated organs and tissues, as follows:

$$H_{E,50} = \sum_T w_T H_{T,50}$$

H_E and $H_{E,50}$ thus reflect both the distribution of dose among the various organs and tissues of the body and their assumed relative sensitivities to stochastic effects. The organ and tissue weighting factor values w_T are as follows: Gonads, 0.25; Breast, 0.15; Red Marrow, 0.12; Lungs, 0.12; Thyroid, 0.03; Bone Surface, 0.03; and Remainder, 0.30 (i.e., a value of $w_T = 0.06$ is applicable to each of the five remaining organs or tissues receiving the highest doses).

The dose delivered to tissues from radiations external to the body occurs only while the radiation field is present. However, the dose delivered to body tissues due to radiations from systemically incorporated radionuclides may continue long after intake of the nuclide has ceased. Therefore, internal doses to specific tissues and organs are typically reported in terms of the committed dose equivalent ($H_{T,50}$), which is defined as the integral of the dose equivalent in a particular tissue T for 50 years after intake (corresponding to a working lifetime).

When subjected to equal doses of radiation, organs and tissues in the human body will exhibit different cancer induction rates. To account for these differences and to normalize radiation doses and effects on a whole body basis for regulation of occupational exposure, the ICRP developed the concept of the effective dose equivalent (H_E) and committed effective dose equivalent ($H_{E,50}$), which are defined as weighted sums of the organ-specific dose equivalents (i.e., $\Sigma\ w_T H_T$) and organ-specific committed dose equivalents (i.e., $\Sigma w_T H_{T,50}$), respectively. Weighting factors, w_T, are based on selected stochastic risk factors specified by the ICRP and are used to average organ-specific dose equivalents (ICRP 1977, 1979). The effective dose equivalent is equal to that dose equivalent, delivered at a uniform whole-body rate, that corresponds to the same number (but possibly a dissimilar distribution) of fatal stochastic health effects as the particular combination of committed organ dose equivalents (see the box on this page).

A special unit, the working level (WL), is used to describe exposure to the short-lived radioactive decay products of radon (Rn-222). Radon is a naturally occurring radionuclide that is of particular concern because it is ubiquitous, it is very mobile in the environment, and it decays through a series of short-lived decay products that can deliver a significant dose to the lung when inhaled. The WL is defined as any combination of short-lived radon decay products in one liter of air that will result in the ultimate emission of 1.3×10^5 MeV of alpha energy. The working level month (WLM) is defined as the exposure to 1 WL for 170 hours (1 working month).

Radiation protection philosophy encourages the reduction of all radiation exposures as low as reasonably achievable (ALARA), in consideration of technical, economic, and social factors. Further, no practice involving radiation exposure should be adopted unless it provides a positive net benefit. In addition to these general guidelines, specific upper limits on radiation exposures and doses have been established by regulatory authorities as described in the following section.

Additional discussion on the measurement of radioactivity is provided in Sections 10.3 and 10.4, and the evaluation of radiation exposure and dose is discussed further in Section 10.5. Discussion of potential health impacts from ionizing radiation is presented in Section 10.6.

10.2 REGULATION OF RADIOACTIVELY CONTAMINATED SITES

Chapter 2 briefly describes the statutes, regulations, guidance, and studies related to the human health evaluation process for chemical contaminants. The discussion describes CERCLA, as amended by SARA, and the RI/FS process. Since radionuclides are classified as hazardous substances under CERCLA, this information is also applicable to radioactively contaminated sites. Chapter 2 also introduces the concept of compliance with applicable or relevant and appropriate requirements (ARARs) in federal and state environmental laws as required by SARA. Guidance on potential ARARs for the remediation of radioactively contaminated sites under CERCLA is available in the *CERCLA Compliance with Other Laws Manual* (EPA 1989c). Only a brief summary of regulatory authorities is presented here.

The primary agencies with regulatory authority for the cleanup of radioactively contaminated sites include EPA, the Nuclear Regulatory Commission (NRC), the Department of Energy (DOE), and state agencies. Other federal agencies, including the Department of Transportation (DOT) and Department of Defense (DOD), also have regulatory programs (but more limited) for radioactive materials. Also, national and international scientific advisory organizations provide recommendations related to radiation protection and radioactive waste management, but have no regulatory authority. The following is a brief description of the main functions and areas of jurisdiction of these agencies and organizations.

- EPA's authority to protect public health and the environment from adverse effects of radiation exposure is derived from several statutes, including the Atomic Energy Act, the Clean Air Act, the Uranium Mill Tailings Radiation Control Act (UMTRCA), the Nuclear Waste Policy Act, the Resource Conservation and Recovery Act (RCRA), and CERCLA. EPA's major responsibilities with regard to radiation include the development of federal guidance and standards, assessment of new technologies, and surveillance of radiation in the environment. EPA also has lead responsibility in the federal government for advising all federal agencies on radiation standards. EPA's radiation standards apply to many different types of activities involving all types of radioactive material (i.e., source, byproduct, special nuclear, and naturally occurring and accelerator produced radioactive material [NARM]). For some of the EPA standards, implementation and enforcement responsibilities are vested in other agencies, such as NRC and DOE.

- NRC licenses the possession and use of certain types of radioactive material at certain types of facilities. Specifically, the NRC is authorized to license source, byproduct, and special nuclear material. The NRC is not authorized to license NARM, although NARM may be partially subject to NRC regulation when it is associated with material licensed by the NRC. Most of DOE's operations are exempt from NRC's licensing and regulatory requirements, as are certain DOD activities involving nuclear weapons and the use of nuclear reactors for military purposes.

- DOE is responsible for conducting or overseeing radioactive material operations at numerous government-owned/contractor-operated facilities. DOE is also responsible for managing several inactive sites that contain radioactive waste, such as sites associated with the Formerly Utilized Sites Remedial Action Program (FUSRAP), the Uranium Mill Tailings Remedial Action Program (UMTRAP), the Grand Junction Remedial

MAJOR FEDERAL LAWS FOR RADIATION PROTECTION

- Atomic Energy Act of 1954, Public Law 83-703 - established the Atomic Energy Commission as the basic regulatory authority for ionizing radiation.
- Energy Reorganization Act of 1974, Public Law 93-438 - amended the Atomic Energy Act, and established the Nuclear Regulatory Commission to regulate nondefense nuclear activities.
- Marine Protection, Research, and Sanctuaries Act of 1972, Public Law 92-532 - established controls for ocean disposal of radioactive waste.
- Safe Drinking Water Act, Public Law 93-523 - mandated regulation of radionuclides in drinking water.
- Clean Air Act Amendments of 1977, Public Law 95-95 - extended coverage of the Act's provisions to include radionuclides.
- Uranium Mill Tailings Radiation Control Act of 1978, Public Law 96-415 - required stabilization and control of byproduct materials (primarily mill tailings) at licensed commercial uranium and thorium processing sites.
- Low-Level Radioactive Waste Policy Act of 1980, Public Law 96-573 - made states responsible for disposal of LLRW generated within their borders and encouraged formation of inter-state compacts.
- Nuclear Waste Policy Act of 1982, Public Law 97-425 - mandated the development of repositories for the disposal of high-level radioactive waste and spent nuclear fuel.
- Low-Level Radioactive Waste Policy Act Amendments of 1985, Public Law 99-240 - amended LLRWPA requirements and

Action Program (GJRAP), and the Surplus Facilities Management Program (SFMP). DOE is authorized to control all types of radioactive materials at sites within its jurisdiction.

- Other federal agencies with regulatory programs applicable to radioactive waste include DOT and DOD. DOT has issued regulations that set forth packaging, labeling, record keeping, and reporting requirements for the transport of radioactive material (see 49 CFR Parts 171 through 179). Most of DOD's radioactive waste management activities are regulated by NRC and/or EPA. However, DOD has its own program for controlling wastes generated for certain nuclear weapon and reactor operations for military purposes. Other agencies, such as the Federal Emergency Management Agency (FEMA) and the Department of the Interior (DOI), may also play a role in radioactive waste cleanups in certain cases.
- States have their own authority and regulations for managing radioactive material and waste. In addition, 29 states (Agreement States) have entered into agreements with the NRC, whereby the Commission has relinquished to the states its regulatory authority over source, byproduct, and small quantities of special nuclear material. Both Agreement States and Nonagreement States can also regulate NARM. Such state-implemented regulations are potential ARARs.
- The National Council on Radiation Protection and Measurements (NCRP) and the International Commission on Radiological Protection (ICRP) provide recommendations on human radiation protection. The NCRP was chartered by Congress to collect, analyze, develop, and disseminate information and recommendations about radiation protection and measurements. The ICRP's function is basically the same, but on an international level. Although neither the NCRP nor the ICRP have regulatory authority, their recommendations serve as the basis for many of the general (i.e., not

source-specific) regulations on radiation protection developed at state and federal levels.

The standards, advisories, and guidance of these various groups are designed primarily to be consistent with each other, often overlapping in scope and purpose. Nevertheless, there are important differences between agencies and programs in some cases. It is important that these differences be well understood so that when more than one set of standards is potentially applicable to or relevant and appropriate for the same CERCLA site, RPMs will be able to evaluate which standards to follow. In general, determination of an ARAR for a site contaminated with radioactive materials requires consideration of the radioactive constituents present and the functional operations that generated the site, whose regulatory jurisdiction the site falls under, and which regulation is most protective, or if relevant and appropriate, most appropriate given site conditions.

For further information on radiation standards, advisories, and guidance, RPMs should consult the detailed ARARs guidance document (EPA 1989c), as well as EPA's ORP and/or Regional Radiation Program Managers.

10.3 DATA COLLECTION

Data collection needs and procedures for sites contaminated with radioactive substances are very similar to those described in Chapter 4 for chemically contaminated sites. There are, however, some basic differences that simplify data collection for radionuclides, including the relative ease and accuracy with which natural background radiation and radionuclide contaminants can be detected in the environment when compared with chemical contaminants.

The pathways of exposure and the mathematical models used to evaluate the potential health risks associated with radionuclides in the environment are similar to those used for evaluating chemical contaminants. Many of the radionuclides found at Superfund sites behave in the environment like trace metals. Consequently, the types of data needed for a radiation risk assessment are very similar to those required for a chemical contaminant risk assessment. For example, the environmental, land use, and demographic data needed and the procedures used to gather the data required to model fate and effect are virtually identical. The primary differences lie in the procedures used to characterize the radionuclide contaminants. In the sections that follow, emphasis is placed on the procedures used to characterize the radionuclide contaminants and not the environmental setting that affects their fate and effects, since the latter has been thoroughly covered in Chapter 4.

10.3.1 RADIATION DETECTION METHODS

Field and laboratory methods used to identify and quantify concentrations of radionuclides in the environment are, in many cases, more exact, less costly, and more easily implemented than those employed for chemical analyses. Selection of a radiometric method depends upon the number of radionuclides of interest, their activities and types of radiations emitted, as well as on the level of sensitivity required and the sample size available. In some cases, the selection process requires prior knowledge of the nature and extent of radioactive contamination present onsite. See the references provided in the box on page 10-12 for detailed guidance on sample collection and preparation, radiochemical procedures, and radiation counters and measurement techniques. The following discussion provides an overview of a few of the radiation detection techniques and instruments currently used to characterize sites contaminated with radioactive materials.

Field methods utilize instrumental techniques rather than radiochemical procedures to determine in-situ identities and concentrations of radionuclides, contamination profiles, and external beta/gamma exposure rates. Field instruments designed for radiation detection (see Exhibit 10-2) are portable, rugged, and relatively insensitive to wide fluctuations in temperature and humidity. At the same time, they are sensitive enough to discriminate between variable levels of background radiation from naturally occurring radionuclides and excess radiation due to radioactive waste. Because of the harsh conditions in which they are sometimes

EXHIBIT 10-2

TYPES OF FIELD RADIATION DETECTION INSTRUMENTS

Instruments	Range of Counting Rate and Other Characteristics	Typical Uses	Remarks
Beta-Gamma Surface Monitors[a]			
Portable Count Rate Meter (Thin Walled or Thin Window G-M Counter)	0-1,000; 0-10,000; 0-100,000 count/min	Surfaces, hands, clothing	Simple, reliable, battery powered
Alpha Surface Monitors			
Portable Air Proportional Counter with Probe	0-100,000 count/min over 100 cm^2	Surfaces, hands, clothing	Not accurate in high humidity; battery powered; fragile window
Portable Gas Flow Counter with Probe	0-100,000 count/min over 100 cm^2	Surfaces, hands, clothing	Not affected by the humidity; battery powered; fragile window
Portable Scintillation Counter with Probe	0-100,000 count/min over 100 cm^2	Surfaces, hands, clothing	Not affected by the humidity; battery powered; fragile window
Air Monitors			
Particle Samplers			
Filter Paper (High-volume)	40 ft^3/min (1.1 m^3/min)	For quick grab samples	Used intermittently; requires separate counter
Filter Paper (Low-volume)	0.1 to 10 ft^3/min (0.003-0.3 m^3/min)	For continuous room air breathing zone monitoring	Used continuously; requires separate counter
Electrostatic Precipitator	3 ft^3/min (0.09 m^3/min)	For continuous monitoring	Sample deposited on cyclindrical shell; requires separate counter
Impinger	20 to 40 ft^3/min (0.6-1.1 m^3/min)	Alpha contamination	Special uses; requires separate counter
Tritium Monitors			
Flow ionization chambers	0.10 pCi/ m^3/min	Continuous monitoring	May be sensitive to other sources of ionization

[a] None of these surface monitors is suitable for tritium detection.

Source: NCRP Report No. 57 (NCRP 1978).

RADIONUCLIDE MEASUREMENT PROCEDURES

Environmental Radiation Measurements (NCRP 1976)

Instrumentation and Monitoring Methods for Radiation Protection (NCRP 1978)

Radiochemical Analytical Procedures for Analysis of Environmental Samples (EPA 1979a)

Eastern Environmental Radiation Facility Radiochemistry Procedures Manual (EPA 1984a)

A Handbook of Radioactivity Measurement Procedures (NCRP 1985a)

operated, and because their detection efficiency varies with photon energy, all field instruments should be properly calibrated in the laboratory against National Bureau of Standards (NBS) radionuclide sources prior to use in the field. Detector response should also be tested periodically in the field against NBS check-sources of known activity.

Commonly used gamma-ray survey meters include Geiger-Muller (G-M) probes, sodium iodide (NaI(Tl)) crystals, and solid-state germanium diodes (Ge(Li)) coupled to ratemeters, scalers, or multichannel analyzers (MCAs). These instruments provide measurements of overall exposure rates in counts per minute, or microRoentgens or microrem per hour. However, only NaI and Ge(Li) detectors with MCAs provide energy spectra of the gamma rays detected and can therefore verify the identity of specific radionuclides. Thin window G-M detectors and Pancake (ionization) probes are used to detect beta particles. Alpha-particle surface monitors include portable air proportional, gas proportional, and zinc sulfide (ZnS) scintillation detectors, which all have very thin and fragile windows. The references in the box on this page provide additional information on several other survey techniques and instruments, such as aerial gamma surveillance used to map gamma exposure rate contours over large areas.

Laboratory methods involve both chemical and instrumental techniques to quantify low-levels of radionuclides in sample media. The preparation of samples prior to counting is an important consideration, especially for samples containing alpha- and beta-emitting radionuclides that either do not emit gamma rays or emit gamma rays of low abundance. Sample preparation is a multistep process that achieves the following three objectives: (1) the destruction of the sample matrix (primarily organic material) to reduce alpha- and beta-particle self-absorption; (2) the separation and concentration of radionuclides of interest to increase resolution and sensitivity; and (3) the preparation of the sample in a suitable form for counting. Appropriate radioactive tracers (i.e., isotopes of the radionuclides of interest that are not present in the sample initially, but are added to the sample to serve as yield determinants) must be selected and added to the sample before a radiochemical procedure is initiated.

For alpha counting, samples are prepared as thin-layer (low mass) sources on membrane filters by coprecipitation with stable carriers or on metal discs by electrodeposition. These sample filters and discs are then loaded into gas proportional counters, scintillation detectors, or alpha spectrometry systems for measurement (see Exhibit 10-3). In a proportional counter, the sample is immersed in a counting gas, usually methane and argon, and subjected to a high voltage field: alpha emissions dissociate the counting gas creating an ionization current proportional to the source strength, which is then measured by the system electronics. In a scintillation detector, the sample is placed in contact with a ZnS phosphor against the window of a photomultiplier (PM) tube: alpha particles induce flashes of light in the phosphor that are converted to an electrical current in the PM tube and measured. Using alpha spectrometry, the sample is placed in a holder in an evacuated chamber facing a solid-state, surface-barrier detector: alpha particles strike the detector and cause electrical impulses, which are sorted by strength into electronic bins and counted. All three systems yield results in counts per minute, which are then converted into activity units using detector- and radionuclide-specific calibration

EXHIBIT 10-3

TYPES OF LABORATORY RADIATION DETECTION INSTRUMENTS [a]

Type of Instrument	Typical Activity Range (mCi)	Typical Sample Form	Data Acquisition and Display
Gas Proportional Counters	10^{-7} to 10^{-3}	Film disc mount, gas	Ratemeter or scaler
Liquid-Scintillation Counters	10^{-7} to 10^{-3}	Up to 20 ml of liquid gel	Accessories for background subtraction, quench correction, internal standard, sample comparison
NaI (T1) Cylindrical or Well Crystals	10^{-6} to 10^{-3}	Liquid, solid, or contained gas, < 4 ml	Ratemeter
			Discriminators for measuring various energy regions
			Multichannel analyzer, or computer plus analog-to-digital converter
			Computational accessories for full-energy-peak identification, quantification, and spectrum stripping
Ionization Chambers	10^{-2} to 10^{3}	Liquid, solid, or contained gas, (can be large in size)	Ionization-current measurement; digital (mCi) readout, as in dose calibrators
Solid-state Detectors	10^{-2} to 10	Various	Multichannel analyzer or computer with various readout options

[a] Source: NCRP Report No. 58 (NCRP 1985a).

values. Alpha spectrometry is the only system, however, that can be used to identify specific alpha-emitting radionuclides.

For beta counting, samples are prepared both as thin-sources and as solutions mixed with scintillation fluid, similar in function to a phosphor. Beta-emitting sources are counted in gas proportional counters at higher voltages than those applied for alpha counting or in scintillation detectors using phosphors specifically constructed for beta-particle detection. Beta-emitters mixed with scintillation fluid are counted in 20 ml vials in beta-scintillation counters: beta-particle interactions with the fluid produce detectable light flashes. Like alpha detectors, beta detectors provide measurements in counts per minute, which are converted to activity units using calibration factors. It should be noted, however, that few detection systems are available for determining the identity of individual beta-emitting radionuclides, because beta particles are emitted as a continuous spectrum of energy that is difficult to characterize and ascribe to any specific nuclide.

It is advisable to count all samples intact in a known geometry on a NaI or Ge(Li) detector system prior to radiochemical analysis, because many radionuclides that emit gamma rays in sufficient abundance and energy can be detected and measured by this process. Even complex gamma-ray spectra emitted by multiple radionuclide sources can be resolved using Ge(Li) detectors, MCAs, and software packages, and specific radionuclide concentrations can be determined. If the sample activity is low or if gamma rays are feeble, then more rigorous alpha or beta analyses are advised.

10.3.2 REVIEWING AVAILABLE SITE INFORMATION

In Chapter 4, reference is made to reviewing the site data for chemical contaminants in accordance with Stage 1 of the Data Quality Objectives (DQO) process (see box on Page 4-4). This process also applies to radionuclides. For further guidance on the applicability of DQOs to radioactively contaminated sites, consult EPA's Office of Radiation Programs.

10.3.3 ADDRESSING MODELING PARAMETER NEEDS

Exhibits 4-1 and 4-2 describe the elements of a conceptual model and the types of information that may be obtained during a site sampling investigation. These exhibits apply to radioactively contaminated sites with only minor modifications. For example, additional exposure pathways for direct external exposure from immersion in contaminated air or water or from contaminated ground surfaces may need to be addressed for certain radionuclides; these exposure pathways are discussed further in subsequent sections. In addition, several of the parameters identified in these exhibits are not as important or necessary for radiological surveys. For example, the parameters that are related primarily to the modeling of organic contaminants, such as the lipid content of organisms, are typically not needed for radiological assessments.

10.3.4 DEFINING BACKGROUND RADIATION SAMPLING NEEDS

As is the case with a chemically contaminated site, the background characteristics of a radioactively contaminated site must be defined reliably in order to distinguish natural background radiation and fallout from the onsite sources of radioactive waste. With the possible exception of indoor sources of Rn-222, it is often possible to make these distinctions because the radiation detection equipment and analytical techniques used are very precise and sensitive. At a chemically contaminated site, there can be many potential and difficult-to-pinpoint offsite sources for the contamination found onsite, confounding the interpretation of field measurements. With a radioactively contaminated site, however, this is not usually a problem because sources of radionuclides are, in general, easier to isolate and identify. In fact, some radionuclides are so specifically associated with particular industries that the presence of a certain radioactive contaminant sometimes acts as a "fingerprint" to identify its source. Additional information on the sources of natural background and man-made radiation in the environment may be found in the references listed in the box on the next page.

NATURAL BACKGROUND RADIATION

Tritium in the Environment (NCRP 1979)

Ionizing Radiation: Sources and Effects (UNSCEAR 1982)

Exposure from the Uranium Series with Emphasis on Radon and its Daughters (NCRP 1984b)

Carbon-14 in the Environment (NCRP 1985c)

Environmental Radioactivity (Eisenbud 1987)

Population Exposure to External Natural Radiation Background in the United States (EPA 1987a)

Ionizing Radiation Exposure of the Population of the United States (NCRP 1987a)

Exposure of the Population of the United States and Canada from Natural Background Radiation (NCRP 1987b)

10.3.5 PRELIMINARY IDENTIFICATION OF POTENTIAL EXPOSURE

Identification of environmental media of concern, the types of radionuclides expected at a site, areas of concern (sampling locations), and potential routes of radionuclide transport through the environment is an important part of the radiological risk assessment process. Potential media of concern include soil, ground water, surface water, air, and biota, as discussed in Chapter 4. Additional considerations for radioactively contaminated sites are listed below.

- Usually a very limited number of radionuclides at a site contribute significantly to the risk. During the site scoping meeting, it is appropriate to consult with a health physicist not only to develop a conceptual model of the facility, but also to identify the anticipated critical radionuclides and pathways.

- In addition to the environmental media identified for chemically contaminated sites, radioactively contaminated sites should be examined for the potential presence of external radiation fields. Many radionuclides emit both beta and gamma radiation, which can create significant external exposures.

- There are other components in the environment that may or may not be critical exposure pathways for the public, but that are very useful indicators of the extent and type of contamination at a site. These components include sediment, aquatic plants, and fish, which may concentrate and integrate the radionuclide contaminants that may be (or have been) present in the aquatic environment at a site. Accordingly, though some components of the environment may or may not be important direct routes of exposure to man, they can serve as indicators of contamination.

10.3.6 DEVELOPING A STRATEGY FOR SAMPLE COLLECTION

The discussions in Chapter 4 regarding sample location, size, type, and frequency apply as well to radioactively contaminated sites with the following additions and qualifications. First, the resolution and sensitivity of radioanalytical techniques permit detection in the environment of most radionuclides at levels that are well below those that are considered potentially harmful. Analytical techniques for nonradioactive chemicals are usually not this sensitive.

For radionuclides, continuous monitoring of the site environment is important, in addition to the sampling and monitoring programs described in Chapter 4. Many field devices that measure external gamma radiation, such as continuous radon monitors and high pressure ionization chambers, provide a real time continuous record of radiation exposure levels and radionuclide concentrations. Such devices are useful for determining the temporal variation of radiation levels at a contaminated site and for comparing these results to the variability observed at background locations. Continuous measure-ments

provide an added level of resolution for quantifying and characterizing radiological risk.

Additional factors that affect the frequency of sampling for radionuclides, besides those discussed in Chapter 4, include the half-lives and the decay products of the radionuclides. Radionuclides with short half-lives, such as Fe-59 (half-life = 44.5 days), have to be sampled more frequently because relatively high levels of contamination can be missed between longer sampling intervals. The decay products of the radionuclides must also be considered, because their presence can interfere with the detection of the parent nuclides of interest, and because they also may be important contributors to risks.

10.3.7 QUALITY ASSURANCE AND QUALITY CONTROL (QA/QC) MEASURES

The QA/QC concepts described in Chapter 4 also apply to sampling and analysis programs for radionuclides, although the procedures differ. Guidance regarding sampling and measurement of radionuclides and QA/QC protocols for their analyses are provided in the publications listed in the box on this page.

The QA/QC protocols used for radionuclide analysis were not developed to meet the evidential needs of the Superfund program; however, it is likely that many of the current radiological QA/QC guidance would meet the intent of Superfund requirements. Some areas where radiological QA/QC guidance may not meet the intent of Superfund are listed below.

- The degree of standardization for radiochemical procedures may be less rigorous in the QA/QC protocols than that required for chemical labs under the Contract Laboratory Program (CLP). In radiochemical laboratories, several different techniques may be used to analyze for a specific radionuclide in a given matrix with comparable results. The CLP requires all participating chemical laboratories to use standardized techniques.
- The required number and type of QC blanks are fewer for radionuclide samples. For example, a "trip" blank is not generally used because radionuclide samples are less likely to be contaminated from direct exposure to air than are samples of volatile organics.

Limited guidance is available that specifies field QA/QC procedures (see the box on this page). These and other issues related to QA/QC guidance for radiological analyses are discussed further in the Section 10.4.

RADIONUCLIDE MEASUREMENT QA/QC PROCEDURES

Quality Control for Environmental Measurements Using Gamma-Ray Spectrometry (EPA 1977b)

Quality Assurance Monitoring Programs (Normal Operation) - Effluent Streams and the Environment (NRC 1979)

Upgrading Environmental Radiation Data (EPA 1980)

Handbook of Analytical Quality Control in Radioanalytical Laboratories (EPA 1987b)

QA Procedures for Health Labs Radiochemistry (American Public Health Association 1987)

10.4 DATA EVALUATION

Chapter 5 describes the procedures for organizing and evaluating data collected during a site sampling investigation for use in risk assessment. The ten-step process outlined for chemical data evaluation is generally applicable to the evaluation of radioactive contaminants, although many of the details must be modified to accommodate differences in sampling and analytical methods.

10.4.1 COMBINING DATA FROM AVAILABLE SITE INVESTIGATIONS

All available data for the site should be gathered for evaluation and sorted by environmental medium sampled, analytical methods, and sampling periods. Decisions should be made, using the process described in Section 5.1, to combine, evaluate individually, or eliminate specific data for use in the quantitative risk assessment.

10.4.2 EVALUATING ANALYTICAL METHODS

As with chemical data, radiological data should be grouped according to the types of analyses performed to determine which data are appropriate for use in quantitative risk assessment. Analytical methods for measuring radioactive contaminants differ from those for measuring organic and inorganic chemicals. Standard laboratory procedures for radionuclide analyses are presented in references, such as those listed in the box on page 10-12. Analytical methods include alpha, beta, and gamma spectrometry, liquid scintillation counting, proportional counting, and chemical separation followed by spectrometry, depending on the specific radionuclides of interest.

Laboratory accreditation procedures for the analysis of radionuclides also differ. Radionuclide analyses are not currently conducted as part of the Routine Analytical Services (RAS) under the Superfund CLP. However, these analyses may be included under Special Analytical Services (SAS). The EPA Environmental Radioactivity Intercomparison Program, coordinated by the Nuclear Radiation Assessment Division of the Environmental Monitoring Systems Laboratory in Las Vegas (EMSL-LV), provides quality assurance oversight for participating radiation measurement laboratories (EPA 1989b). Over 300 federal, state, and private laboratories participate in some phase of the program, which includes analyses for a variety of radionuclides in media (e.g., water, air, milk, and food) with activity concentrations that approximate levels that may be encountered in the environment. Similar intercomparison programs for analysis of thermoluminescent dosimeters (TLDs) for external radiation exposure rate measurements are conducted by the DOE Environmental Measurements Laboratory (EML) and the DOE Radiological and Environmental Services Laboratory (RESL).

In both cases, these intercomparison programs are less comprehensive than the CLP in terms of facility requirements other than analysis of performance evaluation samples, such as laboratory space and procedural requirements, instrumentation, training, and quality control. However, until such time as radiation measurements become fully incorporated in the CLP, use of laboratories that successfully participate in these intercomparison studies may be the best available alternative for ensuring high-quality analytical data. Regardless of laboratory accreditation, all analytical results should be carefully scrutinized and not accepted at face value.

As discussed in Chapter 5 for chemical analyses, radioanalytical results that are not specific for a particular radionuclide (e.g., gross alpha, gross beta) may have limited usefulness for quantitative risk assessment. They can be useful as a screening tool, however. External gamma exposure rate data, although thought of as a screening measurement, can be directly applied as input data for a quantitative risk assessment.

10.4.3 EVALUATING QUANTITATION LIMITS

Lower limits of detection (LLDs), or quantitation limits, for standard techniques for most radionuclide analyses are sufficiently low to ensure the detection of nuclides at activity concentrations well below levels of concern. There are exceptions, however: some radionuclides with very low specific activities, long half-lives, and/or low-energy decay emissions (e.g., I-129, C-14) are difficult to detect precisely using standard techniques. To achieve lower LLDs, a laboratory may: (1) use more sensitive measurement techniques and/or chemical extraction procedures; (2) analyze larger sample sizes; or (3) increase the counting time of the sample. A laboratory may also choose to apply all three options to increase detection capabilities. Exhibit 10-4 presents examples of typical LLDs using standard analytical techniques. The same special considerations noted for chemical analyses

EXHIBIT 10-4

EXAMPLES OF LOWER LIMITS OF DETECTION (LLD) FOR SELECTED RADIONUCLIDES USING STANDARD ANALYTICAL METHODS[a]

Isotope	Sample Media[b]	LLD pCi	LLD Bq	Methodology
Co-60	-Water	10	0.4	Gamma Spectrometry
	-Soil (dry wt.)	0.1	0.004	Gamma Spectrometry
	-Biota (wet wt.)[c]	0.1	0.004	Gamma Spectrometry
	-Air[d]	25	0.9	Gamma Spectrometry
Sr-90	-Water	1	0.04	Radiochemistry
Cs-137	-Water	10	0.4	Gamma Spectrometry
		0.3	0.01	Radiochemistry
	-Soil (dry wt.)	1	0.04	Gamma Spectrometry
		0.3	0.01	Radiochemistry
	-Biota (wet wt.)	1	0.04	Gamma Spectrometry
		0.3	0.01	Radiochemistry
	-Air	30	1	Gamma Spectrometry
Pb-210	-Water	0.2	0.007	Radiochemistry
	-Soil (dry wt.)	0.2	0.007	Radiochemistry
	-Biota (wet wt.)	0.2	0.007	Radiochemistry
	-Air	5	0.2	Radiochemistry
Ra-226	-Water	100	4	Gamma Spectrometry
		0.1	0.004	Radiochemistry
		0.1	0.004	Radon Daughter Emanation
	-Soil (dry wt.)	0.1	0.004	Radon Daughter Emanation
	-Biota (wet wt.)	0.1	0.004	Radon Daughter Emanation
	-Air	1	0.04	Alpha Spectrometry
Th-232	-Water	0.02	0.0007	Alpha Spectrometry
	-Soil (dry wt.)	0.2	0.007	Radiochemistry
	-Biota (wet wt.)	0.02	0.0007	Alpha Spectrometry
	-Air	0.3	0.01	Alpha Proportional Counter
U-234	-Water	0.02	0.0007	Alpha Spectrometry
U-235	-Soil (dry wt.)	0.1	0.004	Alpha Spectrometry
U-238	-Biota (wet wt.)	0.01	0.0004	Alpha Spectrometry
	-Air	0.2	0.007	Alpha Spectrometry

(continued)

EXHIBIT 10-4 (continued)

EXAMPLES OF LOWER LIMITS OF DETECTION (LLD)
FOR SELECTED RADIONUCLIDES USING STANDARD ANALYTICAL METHODS[a]

Isotope	Sample Media[b]	LLD pCi	LLD Bq	Methodology
Pu-238	-Water	0.02	0.0007	Alpha Spectrometry
Pu-239	-Soil (dry wt.)	0.1	0.004	Alpha Spectrometry
Pu-240	-Biota (wet wt.)	0.01	0.0004	Alpha Spectrometry
	-Air	0.2	0.007	Alpha Spectrometry

[a] Source: U.S. Environmental Protection Agency Eastern Environmental Radiation Facility (EPA-EERF), Department of Energy Environmental Measurements Laboratory (DOE-EML), and commercial laboratories. Note that LLDs are radionuclide-, media-, sample size-, and laboratory-specific: higher and lower LLDs than those reported above are possible. The risk assessor should request and report the LLDs supplied by the laboratory performing the analyses.

[b] Nominal sample sizes: water (1 liter), soil (1 kg dry wt.), biota (1 kg wet wt.), and air (1 filter sample).

[c] Biota includes vegetation, fish, and meat.

[d] Air refers to a sample of 300 m^3 of air collected on a filter, which is analyzed for the radionuclide of interest.

would also apply for radionuclides that are not detected in any samples from a particular medium, but are suspected to be present at a site. In these cases, three options may be applied: (1) re-analyze the sample using more sensitive methods; (2) use the LLD value as a "proxy" concentration to evaluate the potential risks at the detection limit; or (3) evaluate the possible risk implication of the radionuclide qualitatively. An experienced health physicist should decide which of these three options would be most appropriate.

When multiple radionuclides are present in a sample, various interferences can occur that may reduce the analytical sensitivity for a particular radionuclide. Also, in some areas of high background radioactivity from naturally occurring radionuclides, it may be difficult to differentiate background contributions from incremental site contamination. It may be possible to eliminate such interferences by radiochemical separation or special instrumental techniques.

A sample with activity that is nondetectable should be reported as less than the appropriate sample and radionuclide-specific LLD value. However, particular caution should be exercised when applying this approach to radionuclides that are difficult to measure and possess unusually high detection limits, as discussed previously. In most cases where a potentially important radionuclide contaminant is suspected, but not detected, in a sample, the sample should be reanalyzed using more rigorous radiochemical procedures and more sophisticated detection techniques.

If radionuclide sample data for a site are reported without sample-specific radionuclide quantitation limits, the laboratory conducting the analyses should be contacted to determine the appropriate LLD values for the analytical techniques and sample media.

10.4.4 EVALUATING QUALIFIED AND CODED DATA

Various data qualifiers and codes may be attached to problem data from inorganic and organic chemical analyses conducted under the CLP as shown in Exhibits 5-4 and 5-5. These include laboratory qualifiers assigned by the laboratory conducting the analysis and data validation qualifiers assigned by personnel involved in data validation. These qualifiers pertain to QA/QC problems and generally indicate questions concerning chemical identity, chemical concentration, or both. No corresponding system of qualifiers has been developed for radioanalytical data, although certain of the CLP data qualifiers might be adopted for use in reporting radioanalytical data. The health physicist should define and evaluate any qualifiers attached to data for radionuclide analyses. Based on the discussions in Chapter 5, the references on methods listed above, and professional judgment, the health physicist should eliminate inappropriate data from use in the risk assessment.

10.4.5 COMPARING CONCENTRATIONS DETECTED IN BLANKS WITH CONCENTRATIONS DETECTED IN SAMPLES

The analysis of blank samples (e.g., laboratory or reagent blanks, field blanks, calibration blanks) is an important component of a proper radioanalytical program. Analysis of blanks provides a measure of contamination introduced into a sample during sampling or analysis activities.

The CLP provides guidance for inorganic and organic chemicals that are not common laboratory contaminants. According to this guidance, if a blank contains detectable levels of any uncommon laboratory chemical, site sample results should be considered positive only if the measured concentration in the sample exceeds five times the maximum amount detected in any blank. Samples containing less than five times the blank concentration should be classified as nondetects, and the maximum blank-related concentration should be specified as the quantitation limit for that chemical in the sample. Though they are not considered to be common laboratory contaminants, radionuclides should not be classified as nondetects using the above CLP guidance. Instead, the health physicist should evaluate all active sample preparation and analytical procedures for possible sources of contamination.

10.4.6 EVALUATING TENTATIVELY IDENTIFIED RADIONUCLIDES

Because radionuclides are not included on the Target Compound List (TCL), they may be classified as tentatively identified compounds (TICs) under CLP protocols. In reality, however, radioanalytical techniques are sufficiently sensitive that the identity and quantity of radionuclides of potential concern at a site can be determined with a high degree of confidence. In some cases, spectral or matrix interferences may introduce uncertainties, but these problems usually can be overcome using special radiochemical and/or instrumental methods. In cases where a radionuclide's identity is not sufficiently well-defined by the available data set: (1) further analyses may be performed using more sensitive methods, or (2) the tentatively identified radionuclide may be included in the risk assessment as a contaminant of potential concern with notation of the uncertainty in its identity and concentration.

10.4.7 COMPARING SAMPLES WITH BACKGROUND

It is imperative to select, collect, and analyze an appropriate number of background samples to be able to distinguish between onsite sources of radionuclide contaminants from radionuclides expected normally in the environment. Background measurements of direct radiation and radionuclide concentrations in all media of concern should be determined at sampling locations geologically similar to the site, but beyond the influence of the site. Screening measurements (e.g., gross alpha, beta, and gamma) should be used to determine whether more sensitive radionuclide-specific analyses are warranted. Professional judgment should be used by the health physicist to select appropriate background sampling locations and analytical techniques. The health physicist should also determine which naturally occurring radionuclides (e.g., uranium, radium, or thorium) detected onsite should be eliminated from the quantitative risk assessment. All man-made radionuclides detected in samples collected should, however, be retained for further consideration.

10.4.8 DEVELOPING A SET OF RADIONUCLIDE DATA AND INFORMATION FOR USE IN A RISK ASSESSMENT

The process described in Section 5.8 for selection of chemical data for inclusion in the quantitative risk assessment generally applies for radionuclides as well. One exception is the lack of CLP qualifiers for radionuclides, as discussed previously. Radionuclides of concern should include those that are positively detected in at least one sample in a given medium, at levels significantly above levels detected in blank samples and significantly above local background levels. As discussed previously, the decision to include radionuclides not detected in samples from any medium but suspected at the site based on historical information should be made by a qualified health physicist.

10.4.9 GROUPING RADIONUCLIDES BY CLASS

Grouping radionuclides for consideration in the quantitative risk assessment is generally unnecessary and inappropriate. Radiation dose and resulting health risk is highly dependent on the specific properties of each radionuclide. In some cases, however, it may be acceptable to group different radioisotopes of the same element that have similar radiological characteristics (e.g., Pu-238/239/240, U-235/238) or belong to the same decay series. Such groupings should be determined very selectively and seldom offer any significant advantage.

10.4.10 FURTHER REDUCTION IN THE NUMBER OF RADIONUCLIDES

For sites with a large number of radionuclides detected in samples from one or more media, the risk assessment should focus on a select group of radionuclides that dominate the radiation dose and health risk to the critical receptors. For example, when considering transport through ground water to distant receptors, transit times may be very long; consequently, only radionuclides with long half-lives or radioactive progeny that are formed during transport may be of concern for that exposure pathway. For direct external exposures, high-energy gamma emitters are of principal concern, whereas

alpha-emitters may dominate doses from the inhalation and ingestion pathways. The important radionuclides may differ for each exposure pathway and must be determined on their relative concentrations, half-lives, environmental mobility, and dose conversion factors (see Section 10.5 for discussion of dose conversion factors) for each exposure pathway of interest.

The total activity inventory and individual concentrations of radionuclides at a Superfund site will change with time as some nuclides decay away and others "grow in" as a result of radioactive decay processes. Consequently, it may be important to evaluate different time scales in the risk assessment. For example, at a site where Ra-226 (half-life = 1600 years) is the only contaminant of concern in soil at some initial time, the Pb-210 (half-life = 22.3 years) and Po-210 (half-life = 138 days) progeny will also become dominant contributors to the activity onsite over a period of several hundred years.

10.4.11 SUMMARIZING AND PRESENTING DATA

Presentation of results of the data collection and evaluation process will be generally the same for radionuclides and chemical contaminants. The sample table formats presented in Exhibits 5-6 and 5-7 are equally applicable to radionuclide data, except that direct radiation measurement data should be added, if appropriate for the radionuclides and exposure pathways identified at the site.

10.5 EXPOSURE AND DOSE ASSESSMENT

This section describes a methodology for estimating the radiation dose equivalent to humans from potential exposures to radionuclides through all pertinent exposure pathways at a remedial site. These estimates of dose equivalent may be used for comparison with radiation protection standards and criteria. However, this methodology has been developed for regulation of occupational radiation exposures for adults and is not completely applicable for estimating health risk to the general population. Section 10.7.2, therefore, describes a separate methodology for estimating health risk.

Chapter 6 describes the procedures for conducting an exposure assessment for chemical contaminants as part of the baseline risk assessment for Superfund sites. Though many aspects of the discussion apply to radionuclides, the term "exposure" is used in a fundamentally different way for radionuclides as compared to chemicals. For chemicals, exposure generally refers to the intake (e.g., inhalation, ingestion, dermal exposure) of the toxic chemical, expressed in units of mg/kg-day. These units are convenient because the toxicity values for chemicals are generally expressed in these terms. For example, the toxicity value used to assess carcinogenic effects is the slope factor, expressed in units of risk of lifetime excess cancers per mg/kg-day. As a result, the product of the intake estimate with the slope factor yields the risk of cancer (with proper adjustments made for absorption, if necessary).

Intakes by inhalation, ingestion, and absorption are also potentially important exposure pathways for radionuclides, although radionuclide intake is typically expressed in units of activity (i.e., Bq or Ci) rather than mass. Radionuclides that enter through these internal exposure pathways may become systemically incorporated and emit alpha, beta, or gamma radiation within tissues or organs. Unlike chemical assessments, an exposure assessment for radioactive contaminants can include an explicit estimation of the radiation dose equivalent. As discussed previously in Section 10.1, the dose equivalent is an expression that takes into consideration both the amount of energy deposited in a unit mass of a specific organ or tissue as a result of the radioactive decay of a specific radionuclide, as well as the relative biological effectiveness of the radiations emitted by that nuclide. (Note that the term dose has a different meaning for radionuclides [dose = energy imparted to a unit mass of tissue] than that used in Chapter 6 for chemicals [dose, or absorbed dose = mass penetrating into an organism].)

Unlike chemicals, radionuclides can have deleterious effects on humans without being taken into or brought in contact with the body. This is because high energy beta particles and photons from radionuclides in contaminated air, water, or soil can travel long distances with only minimum attenuation in these media before depositing their energy in human tissues. External radiation exposures can

result from either exposure to radionuclides at the site area or to radionuclides that have been transported from the site to other locations in the environment. Gamma and x-rays are the most penetrating of the emitted radiations, and comprise the primary contribution to the radiation dose from external exposures. Alpha particles are not sufficiently energetic to penetrate the outer layer of skin and do not contribute significantly to the external dose. External exposure to beta particles primarily imparts a dose to the outer layer skin cells, although high-energy beta radiation can penetrate into the human body.

The quantification of the amount of energy deposited in living tissue due to internal and external exposures to radiation is termed radiation dosimetry. The amount of energy deposited in living tissue is of concern because the potential adverse effects of radiation are proportional to energy deposition. The energy deposited in tissues is proportional to the decay rate of a radionuclide, and not its mass. Therefore, radionuclide quantities and concentrations are expressed in units of activity (e.g., Bq or Ci), rather than in units of mass.

Despite the fundamental difference between the way exposures are expressed for radionuclides and chemicals, the approach to exposure assessment presented in Chapter 6 for chemical contaminants largely applies to radionuclide contaminants. Specifically, the three steps of an exposure assessment for chemicals also apply to radionuclides: (1) characterization of the exposure setting; (2) identification of the exposure pathways; and (3) quantification of exposure. However, some of the methods by which these three steps are carried out are different for radionuclides.

10.5.1 CHARACTERIZING THE EXPOSURE SETTING

Initial characterization of the exposure setting for radioactively contaminated sites is virtually identical to that described in Chapter 6. One additional consideration is that, at sites suspected of having radionuclide contamination, a survey should be conducted to determine external radiation fields using any one of a number of field survey instruments (preferably, G-M tubes and NaI(Tl) field detectors) (see Exhibit 10-2). Health and safety plans should be implemented to reduce the possibility of radiation exposures that are in excess of allowable limits.

REFERENCES ON EXPOSURE ASSESSMENT FOR RADIONUCLIDES

Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents (NRC 1977)

Radiological Assessment: A Textbook on Environmental Dose Analysis (Till and Meyer 1983)

Models and Parameters for Environmental Radiological Assessments (Miller 1984)

Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment (NCRP 1984a)

Background Information Document, Draft EIS for Proposed NESHAPS for Radionuclides, Volume I, Risk Assessment Methodology (EPA 1989a)

Screening Techniques for Determining Compliance with Environmental Standards (NCRP 1989)

10.5.2 IDENTIFYING EXPOSURE PATHWAYS

The identification of exposure pathways for radioactively contaminated sites is very similar to that described in Chapter 6 for chemically contaminated sites, with the following additional guidance.

- In addition to the various ingestion, inhalation, and direct contact pathways described in Chapter 6, external exposure to penetrating radiation should also be considered. Potential external exposure pathways to be considered include immersion in contaminated air, immersion in contaminated water, and radiation

exposure from ground surfaces contaminated with beta- and photon-emitting radionuclides.

- As with nonradioactive chemicals, environmentally dispersed radionuclides are subject to the same chemical processes that may accelerate or retard their transfer rates and may increase or decrease their bioaccumulation potentials. These transformation processes must be taken into consideration during the exposure assessment.

- Radionuclides undergo radioactive decay that, in some respects, is similar to the chemical or biological degradation of organic compounds. Both processes reduce the quantity of the hazardous substance in the environment and produce other substances. (Note, however, that biological and chemical transformations can never alter, i.e., either increase or decrease, the radioactivity of a radionuclide.) Radioactive decay products can also contribute significantly to the radiation exposure and must be considered in the assessment.

- Chapter 6 presents a series of equations (Exhibits 6-11 through 6-19) for quantification of chemical exposures. These equations and suggested default variable values may be used to estimate radionuclide intakes as a first approximation, if the equations are modified by deleting the body weight and averaging time from the denominator. However, depending upon the characteristics of the radionuclides of concern, consideration of radioactive decay and ingrowth of radioactive decay products may be important additions, as well as the external exposure pathways.

- Chapter 6 also refers to a number of computer models that are used to predict the behavior and fate of chemicals in the environment. While those models may be suitable for evaluations of radioactive contaminants in some cases, numerous models have been developed specifically for evaluating the transport of radionuclides in the environment and predicting the doses and risks to exposed individuals. In general, models developed specifically for radiological assessments should be used. Such models include, for example, explicit consideration of radioactive decay and ingrowth of radioactive decay products. (Contact ORP for additional guidance on the fate and transport models recommended by EPA.)

10.5.3 QUANTIFYING EXPOSURE: GENERAL CONSIDERATIONS

One of the primary objectives of an exposure assessment is to make a reasonable estimate of the maximum exposure to individuals and critical population groups. The equation presented in Exhibit 6-9 to calculate intake for chemicals may be considered to be applicable to exposure assessment for radionuclides, except that the body weight and averaging time terms in the denominator should be omitted. However, as discussed previously, exposures to radionuclides include both internal and external exposure pathways. In addition, radiation exposure assessments do not end with the calculation of intake, but take the calculation an additional step in order to estimate radiation dose equivalent.

The radiation dose equivalent to specified organs and the effective dose equivalent due to intakes of radionuclides by inhalation or ingestion are estimated by multiplying the amount of each radionuclide inhaled or ingested times appropriate dose conversion factors (DCFs), which represent the dose equivalent per unit intake. As noted previously, the effective dose equivalent is a weighted sum of the dose equivalents to all irradiated organs and tissues, and represents a measure of the overall detriment. *Federal Guidance Report No. 11* (EPA 1988) provides DCFs for each of over 700 radionuclides for both inhalation and ingestion exposures. It is important to note, however, that these DCFs were developed for regulation of occupational exposures to radiation and may not be appropriate for the general population.

Radionuclide intake by inhalation and ingestion is calculated in the same manner as chemical intake

except that it is not divided by body weight or averaging time. For radionuclides, a reference body weight is already incorporated into the DCFs, and the dose is an expression of energy deposited per gram of tissue.

If intake of a radionuclide is defined for a specific time period (e.g., Bq/year), the dose equivalent will be expressed in corresponding terms (e.g., Sv/year). Because systemically incorporated radionuclides can remain within the body for long periods of time, internal dose is best expressed in terms of the committed effective dose equivalent, which is equal to the effective dose equivalent over the 50-year period following intake.

External exposures may be determined by monitoring and sampling of the radionuclide concentrations in environmental media, direct measurement of radiation fields using portable instrumentation, or by mathematical modeling. Portable survey instruments that have been properly calibrated can display dose rates (e.g., Sv/hr), and dose equivalents can be estimated by multiplying by the duration of exposure to the radiation field. Alternatively, measured or predicted concentrations in environmental media may be multiplied by DCFs, which relate radionuclide concentrations on the ground, in air, or in water to external dose rates (e.g., Sv/hr per Bq/m^2 for ground contamination or Sv/hr per Bq/m^3 for air or water immersion).

The dose equivalents associated with external and internal exposures are expressed in identical units (e.g., Sv), so that contributions from all pathways can be summed to estimate the total effective dose equivalent value and prioritize risk from different sources.

In general, radiation exposure assessments need not consider acute toxicity effects. Acute exposures are of less concern for radionuclides than for chemicals because the quantities of radionuclides required to cause adverse effects from acute exposure are extremely large and such levels are not normally encountered at Superfund sites. Toxic effects from acute radiation exposures are possible when humans are exposed to the radiation from large amounts of radioactive materials released during a major nuclear plant accident, such as Chernobyl, or during above-ground weapons detonations. Consequently, the exposure and risk assessment guidance for radionuclides presented in this chapter is limited to situations causing chronic exposures to low levels of radioactive contaminants.

10.5.4 QUANTIFYING EXPOSURE: DETERMINING EXPOSURE POINT CONCENTRATIONS

The preferred method for estimating the concentration of chemical or radioactive contaminants at those places where members of the public may come into contact with them is by direct measurement. However, this will not be possible in many circumstances and it may be necessary, therefore, to use environmental fate and transport models to predict contaminant concentrations. Such modeling would be necessary, for example: (1) when it is not possible to obtain representative samples for all radionuclides of concern; (2) when the contaminant has not yet reached the potential exposure points; and (3) when the contaminants are below the limits of detection but, if present, can still represent a significant risk to the public.

Numerous fate and transport models have been developed to estimate contaminant concentrations in ground water, soil, air, surface water, sediments, and food chains. Models developed for chemical contaminants, such as those discussed in Chapter 6, may also be applied to radionuclides with allowance for radioactive decay and ingrowth of decay products. There are also a number of models that have been developed specifically for radionuclides. These models are similar to the models used for toxic chemicals but have features that make them convenient to use for radionuclide pathway analysis, such as explicit consideration of radioactive decay and daughter ingrowth. Available models for use in radiation risk assessments range in complexity from a series of hand calculations to major computer codes. For example, NRC Regulatory Guide 1.109 presents a methodology that may be used to manually estimate dose equivalents from a variety of exposure pathways (NRC 1977). Examples of computerized radiological assessment models include the AIRDOS-EPA code and the EPA-PRESTO family of codes, which are used extensively by EPA to estimate exposures and doses to populations following atmospheric releases of radionuclides and releases from a low-level waste

disposal facility, respectively. Guidance on selection and use of the various models can be obtained from the EPA Office of Radiation Programs.

Exhibit 6-10, Example of Table Format for Summarizing Exposure Concentrations, may be used for radionuclide contaminants, except that radionuclide concentrations are expressed in terms of activity per unit mass or volume of the environmental medium (e.g., Bq/kg, Bq/L) rather than mass.

10.5.5 QUANTIFYING EXPOSURE: ESTIMATING INTAKE AND DOSE EQUIVALENT

Section 6.6 presents a description of the methods used to estimate intake rates of contaminants from the various exposure pathways. Exhibits 6-11 to 6-19 present the equations and input assumptions recommended for use in intake calculations. In concept, those equations and assumptions also apply generally to radionuclides, except that the body weight and averaging time terms in the denominators should be omitted. However, as discussed previously, the product of these calculations for radionuclides is an estimate of the radionuclide intake, expressed in units of activity (e.g., Bq), as opposed to mg/kg-day. In addition, the endpoint of a radiation exposure assessment is radiation dose, which is calculated using DCFs as explained below. As explained previously, dose equivalents calculated in the following manner should be used to compare with radiation protection standards and criteria, not to estimate risk.

Internal Exposure. Exhibits 6-11, 6-12, 6-14, 6-17, 6-18, and 6-19 present simplified models for the ingestion of water, food, and soil as pathways for the intake of environmental contaminants. The recommended assumptions for ingestion rates and exposure durations are applicable to radionuclide exposures and may be used to estimate the intake rates of radionuclides by these pathways. As noted previously, however, these intake estimates for radionuclides should not be divided by the body weight or averaging time. These intake rates must be multiplied by appropriate DCF values in order to obtain committed effective dose equivalent values. The more rigorous and complex radionuclide pathway models noted previously typically require much more extensive input data and may include default parameter values that differ somewhat from the values recommended in these exhibits.

Exhibit 6-16 presents the equation and assumptions used to estimate the contaminant intake from air. For radionuclides, the dose from inhalation of contaminated air is determined as the product of the radionuclide concentration in air (Bq/m^3), the breathing rate (m^3 per day or year), exposure duration (day or year), and the inhalation DCF (Sv per Bq inhaled). The result of this calculation is the committed effective dose equivalent, in units of Sv.

Chapter 6 points out that dermal absorption of airborne chemicals is not an important route of uptake. This point is also true for most radionuclides, except airborne tritiated water vapor, which is efficiently taken into the body through dermal absorption. In order to account for this route of uptake, the inhalation DCF for tritium includes an adjustment factor to account for dermal absorption.

External Exposure. Immersion in air containing certain beta-emitting and/or photon-emitting radioactive contaminants can also result in external exposures. Effective dose equivalents from external exposure are calculated as the product of the airborne radionuclide concentration (Bq/m^3), the external DCF for air immersion (Sv/hr per Bq/m^3), and the duration of exposure (hours).

Exhibits 6-13 and 6-15 illustrate the dermal uptake of contaminants resulting from immersion in water or contact with soil. This route of uptake can be important for many organic chemicals; however, dermal uptake is generally not an important route of uptake for radionuclides, which have small dermal permeability constants. External radiation exposure due to submersion in water contaminated with radionuclides is possible and is similar to external exposure due to immersion in air. However, because of the shielding effects of water and the generally short durations of such exposures, immersion in water is typically of lesser significance. The product of the radionuclide concentration in water (Bq/m^3), the relevant DCF (Sv/hr per Bq/m^3), and the duration of exposure (hours) yields effective dose equivalent.

The third external exposure pathway of potential significance is irradiation from radionuclides deposited on the ground surface. Effective dose equivalents resulting from this pathway may be estimated as the product of the soil surface concentration (Bq/m^2) of photon-emitting radionuclides of concern, the external DCF for ground surface exposure (Sv/hr per Bq/m^2), and the duration of exposure (hours).

10.5.6 COMBINING INTAKES AND DOSES ACROSS PATHWAYS

The calculations described previously result in estimates of committed effective dose equivalents (Sv) from individual radionuclides via a large number of possible exposure pathways. Because a given population may be subject to multiple exposure pathways, the results of the exposure assessment should be organized by grouping all applicable exposure pathways for each exposed population. Risks from various exposure pathways and contaminants then can be integrated during the risk characterization step (see Section 10.7).

10.5.7 EVALUATING UNCERTAINTY

The radiation exposure assessment should include a discussion of uncertainty, that, at a minimum, should include: (1) a tabular summary of the values used to estimate exposures and doses and the range of these values; and (2) a summary of the major assumptions of the exposure assessment, including the uncertainty associated with each assumption and how it might affect the exposure and dose estimates. Sources of uncertainty that must be addressed include: (1) how well the monitoring data represent actual site conditions; (2) the exposure models, assumptions, and input variables used to estimate exposure point concentrations; and (3) the values of the variables used to estimate intakes and external exposures. More comprehensive discussions of uncertainty associated with radiological risk assessment are provided in the *Background Information Document for the Draft EIS for Proposed NESHAPS for Radionuclides* (EPA 1989a), *Radiological Assessment* (Till and Meyer 1983), and NCRP Report No. 76 (NCRP 1984a).

10.5.8 SUMMARIZING AND PRESENTING EXPOSURE ASSESSMENT RESULTS

Exhibit 6-22 presents a sample format for summarizing the results of the exposure assessment. The format may also be used for radionuclide contaminants except that the entries should be specified as committed effective dose equivalents (Sv) and the annual estimated intakes (Bq) for each radionuclide of concern. The intakes and dose estimates should be tabulated for each exposure pathway so that the most important radionuclides and pathways contributing to the total health risk may be identified.

The information should be organized by exposure pathway, population exposed, and current and future use assumptions. For radionuclides, however, it may not be necessary to summarize short-term and long-term exposures separately as specified for chemical contaminants.

10.6 TOXICITY ASSESSMENT

Chapter 7 describes the two-step process employed to assess the potential toxicity of a given chemical contaminant. The first step, hazard identification, is used to determine whether exposure to a contaminant can increase the incidence of an adverse health effect. The second step, dose-response assessment, is used to quantitatively evaluate the toxicity information and characterize the relationship between the dose of the contaminant administered or received and the incidence of adverse health effects in the exposed population.

There are certain fundamental differences between radionuclides and chemicals that somewhat simplify toxicity assessment for radionuclides. As discussed in the previous sections, the adverse effects of exposure to radiation are due to the energy deposited in sensitive tissue, which is referred to as the radiation dose. In theory, any dose of radiation has the potential to produce an adverse effect. Accordingly, exposure to any radioactive substances is, by definition, hazardous.

Dose-response assessment for radionuclides is also more straightforward. The type of effects and

the likelihood of occurrence of any one of a number of possible adverse effects from radiation exposure depends on the radiation dose. The relationship between dose and effect is relatively well characterized (at high doses) for most types of radiations. As a result, the toxicity assessment, within the context that it is used in this manual, need not be explicitly addressed in detail for individual radionuclides at each contaminated site.

The sections that follow provide a brief summary of the human and experimental animal studies that establish the hazard and dose-response relationship for radiation exposure. More detailed discussions of radiation toxicity are provided in publications of the National Academy of Sciences Committee on Biological Effects of Ionizing Radiation (BEIR), the United Nations Scientific Committee on Effects of Atomic Radiation (UNSCEAR), NRC, NCRP, and ICRP listed in the box on this page.

REFERENCES ON HEALTH EFFECTS OF RADIATION EXPOSURE

Recommendations of the ICRP (ICRP 1977)

Limits for Intake of Radionuclides by Workers (ICRP 1979)

Influence of Dose and Its Distribution in Time on Dose-Response Relationships for Low-LET Radiations (NCRP 1980)

The Effects on Populations of Exposure to Low Levels of Ionizing Radiation (NAS 1980)

Induction of Thyroid Cancer by Ionizing Radiation (NCRP 1985b)

Lung Cancer Risk from Indoor Exposures to Radon Daughters (ICRP 1987)

Health Risks of Radon and Other Internally Deposited Alpha-Emitters (National Academy of Sciences 1988)

Ionizing Radiation: Sources, Effects, and Risks (UNSCEAR 1988)

Health Effects Models for Nuclear Power Plant Accident Consequence Analysis: Low-LET Radiation (NRC 1989)

10.6.1 HAZARD IDENTIFICATION

The principal adverse biological effects associated with ionizing radiation exposures from radioactive substances in the environment are carcinogenicity, mutagenicity, and teratogenicity. Carcinogenicity is the ability to produce cancer. Mutagenicity is the property of being able to induce genetic mutation, which may be in the nucleus of either somatic (body) or germ (reproductive) cells. Mutations in germ cells lead to genetic or inherited defects. Teratogenicity refers to the ability of an agent to induce or increase the incidence of congenital malformations as a result of permanent structural or functional deviations produced during the growth and development of an embryo (more commonly referred to as birth defects). Radiation may induce other deleterious effects at acute doses above about 1 Sv, but doses of this magnitude are not normally associated with radioactive contamination in the environment.

As discussed in Section 10.1, ionizing radiation causes injury by breaking molecules into electrically charged fragments (i.e., free radicals), thereby producing chemical rearrangements that may lead to permanent cellular damage. The degree of biolog--ical damage caused by various types of radiation varies according to how spatially close together the ionizations occur. Some ionizing radiations (e.g., alpha particles) produce high density regions of ionization. For this reason, they are called high-LET (linear energy transfer) particles. Other types of radiation (e.g., x-rays, gamma rays, and beta particles) are called low-LET radiations because of the low density pattern of ionization they produce. In equal doses, the carcinogenicity and mutagenicity of high-LET radiations may be an order of magnitude or more greater than those of low-LET radiations, depending on the endpoint being evaluated. The variability in biological effectiveness is accounted for by the quality factor used to calculate the dose equivalent (see Section 10.1).

Carcinogenesis. An extensive body of literature exists on radiation carcinogenesis in man and animals. This literature has been reviewed most recently by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and the National Academy of Sciences Advisory Committee on the Biological Effects of Ionizing Radiations (NAS-BEIR Committee) (UNSCEAR

1977, 1982, 1988; NAS 1972, 1980, 1988). Estimates of the average risk of fatal cancer from low-LET radiation from these studies range from approximately 0.007 to 0.07 fatal cancers per sievert.

An increase in cancer incidence or mortality with increasing radiation dose has been demonstrated for many types of cancer in both human populations and laboratory animals (UNSCEAR 1982, 1988; NAS 1980, 1988). Studies of humans exposed to internal or external sources of ionizing radiation have shown that the incidence of cancer increases with increased radiation exposure. This increased incidence, however, is usually associated with appreciably greater doses and exposure frequencies than those encountered in the environment. Therefore, risk estimates from small doses obtained over long periods of time are determined by extrapolating the effects observed at high, acute doses. Malignant tumors in various organs most often appear long after the radiation exposure, usually 10 to 35 years later (NAS 1980, 1988; UNSCEAR 1982, 1988). Radionuclide metabolism can result in the selective deposition of certain radionuclides in specific organs or tissues, which, in turn, can result in larger radiation doses and higher-than-normal cancer risk in these organs.

Ionizing radiation can be considered pancarcinogenic, i.e., it acts as a complete carcinogen in that it serves as both initiator and promoter, and it can induce cancers in nearly any tissue or organ. Radiation-induced cancers in humans have been reported in the thyroid, female breast, lung, bone marrow (leukemia), stomach, liver, large intestine, brain, salivary glands, bone, esophagus, small intestine, urinary bladder, pancreas, rectum, lymphatic tissues, skin, pharynx, uterus, ovary, mucosa of cranial sinuses, and kidney (UNSCEAR 1977, 1982, 1988; NAS 1972, 1980, 1988). These data are taken primarily from studies of human populations exposed to high levels of radiation, including atomic bomb survivors, underground miners, radium dial painters, patients injected with thorotrast or radium, and patients who received high x-ray doses during various treatment programs. Extrapolation of these data to much lower doses is the major source of uncertainty in determining low-level radiation risks (see EPA 1989a). It is assumed that no lower threshold exists for radiation carcinogenesis.

On average, approximately 50 percent of all of the cancers induced by radiation are lethal. The fraction of fatal cancers is different for each type of cancer, ranging from about 10 percent in the case of thyroid cancer to 100 percent in the case of liver cancer (NAS 1980, 1988). Females have approximately 2 times as many total cancers as fatal cancers following radiation exposure, and males have approximately 1.5 times as many (NAS 1980).

Mutagenesis. Very few quantitative data are available on radiogenic mutations in humans, particularly from low-dose exposures. Some mutations are so mild they are not noticeable, while other mutagenic effects that do occur are similar to nonmutagenic effects and are therefore not necessarily recorded as mutations. The bulk of data supporting the mutagenic character of ionizing radiation comes from extensive studies of experimental animals (UNSCEAR 1977, 1982, 1988; NAS 1972, 1980, 1988). These studies have demonstrated all forms of radiation mutagenesis, including lethal mutations, translocations, inversions, nondisjunction, and point mutations. Mutation rates calculated from these studies are extrapolated to humans and form the basis for estimating the genetic impact of ionizing radiation on humans (NAS 1980, 1988; UNSCEAR 1982, 1988). The vast majority of the demonstrated mutations in human germ cells contribute to both increased mortality and illness (NAS 1980; UNSCEAR 1982). Moreover, the radiation protection community is generally in agreement that the probability of inducing genetic changes increases linearly with dose and that no "threshold" dose is required to initiate heritable damage to germ cells.

The incidence of serious genetic disease due to mutations and chromosome aberrations induced by radiation is referred to as genetic detriment. Serious genetic disease includes inherited ill health, handicaps, or disabilities. Genetic disease may be manifest at birth or may not become evident until some time in adulthood. Radiation-induced genetic detriment includes impairment of life, shortened life span, and increased hospitalization. The frequency of radiation-induced genetic impairment is relatively small in comparison with the magnitude of detriment associated with spontaneously arising genetic diseases (UNSCEAR 1982, 1988).

Teratogenesis. Radiation is a well-known teratogenic agent. The developing fetus is much more sensitive to radiation than the mother. The age of the fetus at the time of exposure is the most important factor in determining the extent and type of damage from radiation. The malformations produced in the embryo depend on which cells, tissues, or organs in the fetus are most actively differentiating at the time of radiation exposure. Embryos are relatively resistant to radiation-induced teratogenic effects during the later stages of their development and are most sensitive from just after implantation until the end of organogenesis (about two weeks to eight weeks after conception) (UNSCEAR 1986; Brent 1980). Effects on nervous system, skeletal system, eyes, genitalia, and skin have been noted (Brent 1980). The brain appears to be most sensitive during development of the neuroblast (these cells eventually become the nerve cells). The greatest risk of brain damage for the human fetus occurs at 8 to 15 weeks, which is the time the nervous system is undergoing the most rapid differentiation and proliferation of cells (Otake 1984).

10.6.2 DOSE-RESPONSE RELATIONSHIPS

This section describes the relationship of the risk of fatal cancer, serious genetic effects, and other detrimental health effects to exposure to low levels of ionizing radiation. Most important from the standpoint of the total societal risk from exposures to low-level ionizing radiation are the risks of cancer and genetic mutations. Consistent with our current understanding of their origins in terms of DNA damage, these effects are believed to be stochastic; that is, the probability (risk) of these effects increases with the dose of radiation, but the severity of the effects is independent of dose. For neither induction of cancer nor genetic effects, moreover, is there any convincing evidence for a "threshold" (i.e., some dose level below which the risk is zero). Hence, so far as is known, any dose of ionizing radiation, no matter how small, might give rise to a cancer or to a genetic effect in future generations. Conversely, there is no way to be certain that a given dose of radiation, no matter how large, has caused an observed cancer in an individual or will cause one in the future.

Exhibit 10-5 summarizes EPA's current estimates of the risk of adverse effects associated with human exposure to ionizing radiation (EPA 1989a). Important points from this summary table are provided below.

- Very large doses (>1 Sv) of radiation are required to induce acute and irreversible adverse effects. It is unlikely that such exposures would occur in the environmental setting associated with a potential Superfund site.

- The risks of serious noncarcinogenic effects associated with chronic exposure to radiation include genetic and teratogenic effects. Radiation-induced genetic effects have not been observed in human populations, and extrapolation from animal data reveals risks per unit exposure that are smaller than, or comparable to, the risk of cancer. In addition, the genetic risks are spread over several generations. The risks per unit exposure of serious teratogenic effects are greater than the risks of cancer. However, there is a possibility of a threshold, and the exposures must occur over a specific period of time during gestation to cause the effect. Teratogenic effects can be induced only during the nine months of pregnancy. Genetic effects are induced during the 30-year reproductive generation and cancer can be induced at any point during the lifetime. If a radiation source is not controlled, therefore, the cumulative risk of cancer may be many times greater than the risk of genetic or teratogenic effects due to the potentially longer period of exposure.

EXHIBIT 10-5

SUMMARY OF EPA'S RADIATION RISK FACTORS[a]

Risk	Significant Exposure Period	Risk Factor Range
Low LET (Gy^{-1})		
Teratogenic:[b]		
Severe mental retardation	Weeks 8 to 15 of gestation	0.25-0.55
Genetic:		
Severe hereditary defects, all generations	30-year reproductive generation	0.006-0.11
Somatic:		
Fatal cancers	Lifetime	0.012-0.12
	In utero	0.029-0.10
All cancers	Lifetime	0.019-0.19
High LET (Gy^{-1})		
Genetic:		
Severe hereditary defects, all generations	30-year reproductive generation	0.016-0.29
Somatic:		
Fatal cancers	Lifetime	0.096-0.96
All cancers	Lifetime	0.15-1.5
Radon Decay Products (10^{-6} WLM^{-1})		
Fatal lung cancer	Lifetime	140-720

[a] In addition to the stochastic risks indicated, acute toxicity may occur at a mean lethal dose of 3-5 Sv with a threshold in excess of 1 Sv.

[b] The range assumes a linear, non-threshold dose-response. However, it is plausible that a threshold may exist for this effect.

Based on these observations, it appears that the risk of cancer is limiting and may be used as the sole basis for assessing the radiation-related human health risks of a site contaminated with radionuclides.

For situations where the risk of cancer induction in a specific target organ is of primary interest, the committed dose equivalent to that organ may be multiplied by an organ-specific risk factor. The relative radiosensitivity of various organs (i.e., the cancer induction rate per unit dose) differs markedly for different organs and varies as a function of the age and sex of the exposed individual. Tabulations of such risk factors as a function of age and sex are provided in the *Background Information Document for the Draft Environmental Impact Statement for Proposed NESHAPS for Radionuclides* (EPA 1989a) for cancer mortality and cancer incidence.

10.7 RISK CHARACTERIZATION

The final step in the risk assessment process is risk characterization. This is an integration step in which the risks from individual radionuclides and pathways are quantified and combined where appropriate. Uncertainties also are examined and discussed in this step.

10.7.1 REVIEWING OUTPUTS FROM THE TOXICITY AND EXPOSURE ASSESSMENTS

The exposure assessment results should be expressed as estimates of radionuclide intakes by inhalation and ingestion, exposure rates and duration for external exposure pathways, and committed effective dose equivalents to individuals from all relevant radionuclides and pathways. The risk assessor should compile the supporting documentation to ensure that it is sufficient to support the analysis and to allow an independent duplication of the results. The review should also confirm that the analysis is reasonably complete in terms of the radionuclides and pathways addressed.

In addition, the review should evaluate the degree to which the assumptions inherent in the analysis apply to the site and conditions being addressed. The mathematical models used to calculate dose use a large number of environmental transfer factors and dose conversion factors that may not always be entirely applicable to the conditions being analyzed. For example, the standard dose conversion factors are based on certain generic assumptions regarding the characteristics of the exposed individual and the chemical and physical properties of the radionuclides. Also, as is the case for chemical contaminants, the environmental transfer factors used in the models may not apply to all settings.

Though the risk assessment models may include a large number of radionuclides and pathways, the important radionuclides and pathways are usually few in number. As a result, it is often feasible to check the computer output using hand calculations. This type of review can be performed by health physicists familiar with the models and their limitations. Guidance on conducting such calculations is provided in numerous references, including Till and Meyer (1983) and NCRP Report No. 76 (NCRP 1984a).

10.7.2 QUANTIFYING RISKS

Given that the results of the exposure assessment are virtually complete, correct, and applicable to the conditions being considered, the next step in the process is to calculate and combine risks. As discussed previously, the risk assessment for radionuclides is somewhat simplified because only radiation carcinogenesis needs to be considered.

Section 10.5 presents a methodology for estimating committed effective dose equivalents that may be compared with radiation protection standards and criteria. Although the product of these dose equivalents (Sv) and an appropriate risk factor (risk per Sv) yields an estimate of risk, the health risk estimate derived in such a manner is not completely applicable for members of the general public. A better estimate of risk may be computed using age- and sex-specific coefficients for individual organs receiving significant radiation doses. This information may be used along with organ-specific dose conversion factors to derive slope factors that represent the age-averaged lifetime excess cancer incidence per unit intake for the radionuclides of concern. The Integrated Risk Information System (IRIS) contains slope factor values for radionuclides of concern at remedial sites for each of the four

major exposure pathways (inhalation, ingestion, air immersion, and ground-surface irradiation), along with supporting documentation for the derivation of these values (see Chapter 7 for more detail on IRIS).

The slope factors from the IRIS data base for the inhalation pathway should be multiplied by the estimated inhaled activity (derived using the methods presented in Section 6.6.3 and Exhibit 6-16, without division of the body weight and averaging time) for each radionuclide of concern to estimate risks from the inhalation pathway. Similarly, risks from the ingestion pathway should be estimated by multiplying the ingestion slope factors by the activity ingested for each radionuclide of concern (derived using the methods presented in Exhibits 6-11, 6-12, 6-14, 6-17, 6-18, and 6-19, without division by the body weight and averaging time). Estimates of the risk from the air immersion pathway should be computed by multiplying the appropriate slope factors by the airborne radionuclide concentration (Bq/m^3) and the duration of exposure. Risk from the ground surface pathway should be computed as the product of the slope factor, the soil concentration (Bq/m^2), and the duration of exposure for each radionuclide of concern.

The sum of the risks from all radionuclides and pathways yields the lifetime risk from the overall exposure. As discussed in Chapter 8, professional judgment must be used in combining the risks from various pathways, as it may not be physically possible for one person to be exposed to the maximum radionuclide concentrations for all pathways.

10.7.3 COMBINING RADIONUCLIDE AND CHEMICAL CANCER RISKS

Estimates of the lifetime risk of cancer to exposed individuals resulting from radiological and chemical risk assessments may be summed in order to determine the overall potential human health hazard associated with a site. Certain precautions should be taken, however, before summing these risks. First, the risk assessor should evaluate whether it is reasonable to assume that the same individual can receive the maximum radiological and chemical dose. It is possible for this to occur in some cases because many of the environmental transport processes and routes of exposure are the same for radionuclides and chemicals.

In cases where different environmental fate and transport models have been used to predict chemical and radionuclide exposure, the mathematical models may incorporate somewhat different assumptions. These differences can result in incompatibilities in the two estimates of risk. One important difference of this nature is how the cancer toxicity values (i.e., slope factors) were developed. For both radionuclides and chemicals, cancer toxicity values are obtained by extrapolation from experimental and epidemiological data. For radionuclides, however, human epidemiological data form the basis of the extrapolation, while for many chemical carcinogens, laboratory experiments are the primary basis for the extrapolation. Another even more fundamental difference between the two is that slope factors for chemical carcinogens generally represent an upper bound or 95th percent confidence limit value, while radionuclide slope factors are best estimate values.

In light of these limitations, the two sets of risk estimates should be tabulated separately in the final baseline risk assessment.

10.7.4 ASSESSING AND PRESENTING UNCERTAINTIES

Uncertainties in the risk assessment must be evaluated and discussed, including uncertainties in the physical setting definition for the site, in the models used, in the exposure parameters, and in the toxicity assessment. Monte Carlo uncertainty analyses are frequently performed as part of the uncertainty and sensitivity analysis for radiological risk assessments. A summary of the use of uncertainty analyses in support of radiological risk assessments is provided in NCRP Report No. 76 (NCRP 1984a), *Radiological Assessment* (Till and Meyer 1983), and in the *Background Information Document for the Draft EIS for Proposed NESHAPs for Radionuclides* (EPA 1989a).

10.7.5 SUMMARIZING AND PRESENTING THE BASELINE RISK CHARACTERIZATION RESULTS

The results of the baseline risk characterization should be summarized and presented in an effective manner to assist in decision-making. The estimates of risk should be summarized in the context of the specific site conditions. Information should include the identity and concentrations of radionuclides, types and magnitudes of health risks predicted, uncertainties in the exposure estimates and toxicity information, and characteristics of the site and potentially exposed populations. A summary table should be provided in a format similar to that shown in Exhibit 6-22, as well as graphical presentations of the predicted health risks (see Exhibit 8-7).

10.8 DOCUMENTATION, REVIEW, AND MANAGEMENT TOOLS FOR THE RISK ASSESSOR, REVIEWER, AND MANAGER

The discussion provided in Chapter 9 also applies to radioactively contaminated sites. The suggested outline provided in Exhibit 9-1 may also be used for radioactively contaminated sites with only minor modifications. For example, the portions that uniquely pertain to the CLP program and noncarcinogenic risks are not needed. In addition, because radionuclide hazard and toxicity have been addressed adequately on a generic basis, there is no need for an extensive discussion of toxicity in the report.

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